1-1-2012

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Recommended Citation

Jenkins, MJ, Page, WG, Hebertson, EG, Alexander, ME. 2012. Fuels and fire behavior dynamics in bark beetle attacked forests in western North America and implications for fire management. Forest Ecology and Management. 275:23-34.

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Fuels and Fire Behavior Dynamics in Bark Beetle-Attacked Forests in Western North America and Implications for Fire Management

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Abstract

Declining forest health attributed to associations between extensive bark beetle-caused tree mortality, accumulations of hazardous fuels, wildfire, and climate change have catalyzed changes in forest health and wildfire protection policies of land management agencies. These changes subsequently prompted research to investigate the extent to which bark beetle-altered fuel complexes affect fire behavior. Although not yet rigorously quantified, the results of the investigations, in addition to a growing body of operational experience and research, indicates that predictable changes in surface, ladder and canopy fuel characteristics do occur over the course of a bark beetle rotation. Input of these changes in fuel characteristics into conventional fire behavior modeling systems can readily provide predictions of potential fire behavior, including the likelihood of crowning. However, several factors limit the direct application of these modeling systems in their current form and consequently, they may largely under predict fire potential in such stands. This presents a concern where extreme fire behavior involving both crowning and spotting coupled with flammable fuel conditions can pose serious challenges to

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incident management and threaten the safety of firefighters and the general public alike. In this paper, we review the nature and characteristics of bark beetle-altered fuel complexes in the conifer forests of the Interior West and the challenges of understanding the effects on extreme fire behavior, including the initiation and spread of crown fires. We also discuss how emerging fire management plans in the US have begun to integrate wildfire management and other forest health objectives with the specific goal of achieving biodiversity and ecosystem resiliency while simultaneously reducing the existence of hazardous fuel complexes.

Keywords: crown fire, *Dendroctonus*, fire behavior, fire modeling, fire suppression, forest flammability, fuel accumulation, tree mortality

1. Introduction

Agents of disturbance in forested ecosystems include wind, snow, ice, insects, pathogens, fires, avalanches, hurricanes and floods. Many of these disturbances occur in a random, non-cyclic and unpredictable manner while others occur periodically and are largely predictable. The occurrence of disturbance events, their timing, severity, frequency, magnitude, and interactions over time and space characterize a disturbance regime (Pickett and White, 1985). Few forests in western North America are free from the effects of disturbance. To the contrary, biotic and abiotic agents regulate many aspects of forest composition and structure. The interaction of

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disturbance agents over large spatial and long temporal scales often determines the nature of the forested landscape (Veblen *et al.*, 1994). Among biotic agents of disturbance, bark beetles have the ability to dramatically alter stand composition and structure, fuels quantity and quality, and carbon cycling over very short to long time frames (Hawkes *et al.*, 2004; Jenkins *et al.*, 2008; Kurz *et al.*, 2008)

Bark beetles in the genus *Dendroctonus* (Coleoptera: Curculionidae, Scolytinae) are native insects that play an important role in western North American coniferous forest ecosystems. At low population levels bark beetles typically infest large, old, and weakened trees, whose deaths serve to recycle nutrients and create openings for regeneration. Large-scale outbreaks have been a common feature of coniferous forests at least since the last glacial retreat about 13,000 years ago (Brunelle *et al.*, 2008). Bark beetle outbreaks and the associated loss of mature host trees results in a modification of stand and age-class structure and species composition. During the outbreak phase the bark beetle population and host tree mortality increase by orders of magnitude and may impact forest management activities (Teale and Castello, 2011). Warm, dry weather conditions may trigger outbreaks by stressing otherwise vigorous trees and decreasing bark beetle development time (Raffa *et al.*, 2008). Hebertson and Jenkins (2008) showed that spruce beetle (*D. rufipennis* Kirby) outbreaks were associated with prolonged droughts during the past century.

Over the past 25 years a total of 4.3 million hectares of western coniferous forests were infested by bark beetles including 6.6 million hectares by mountain pine beetle, 128,000 hectares by spruce beetle and 185,000 hectares by Douglas-fir beetle (*D. pseudotsugae* Hopkins) (Man, 2010). Mountain pine beetle mortality in the western United States has also increased dramatically in high elevation five-needle pines reaching levels not previously recorded and

resulting in the deaths of six million five-needle pines on 486,000 hectares (Gibson *et al.*, 2008) and the loss of up to 95% of cone bearing white bark pine **(***Pinus albicaulis* Engelm.) trees in some areas. The scale of bark beetle caused tree mortality, particularly in high elevation species such as whitebark pine may be unprecedented and influenced by climate warming (Logan *et al.*, 2010). It is equally important to note, however, that the initiation of bark beetle outbreaks and population spread is not possible without susceptible stands which are usually dense and composed of a large percentage (>60%) of mature, large diameter host trees. Changes to fuels complexes and fire behavior due to 20th century fire suppression and exclusion policies, livestock grazing and a more recent decrease in active timber management have created an abundance of large, old conifers in western North America (Samman and Logan, 2000). The rash of large, human-caused wildfires in the late 1800s in subalpine forests may have also contributed to increased landscape homogeneity by initiating stands that, by the 20th century, were susceptible to bark beetle attack in terms of size and age class distribution (Sibold *et al.*, 2006; Baker, 2009). The period of landscape level fires in subalpine forests may have also been associated with major climate drivers and a generally warmer and drier climate (Kitzberger *et al.*, 2007; Schoennagel *et al.*, 2007).

Fire is the most important abiotic disturbance in western forest ecosystems (Wright and Heinselman, 1973). Fires vary in kind, frequency, and magnitude resulting in a broad range of vegetative responses. For example, crown fires regulate the availability of sites for the initiation of new stands by killing the majority of living trees and exposing mineral soil (White, 1979; Oliver, 1981). Surface fires typically result in partial removal of the overstory or the death of individual trees creating canopy gaps and allowing recruitment of subcanopy trees into the overstory (White *et al.*, 1984; Veblen *et al.*, 1994).

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The number of large fires, the total annual area burned, and associated suppression costs have also risen dramatically across much of the western United States over the past 25 years (Calkin *et al.*, 2005; Stephens, 2005; GAO, 2007). The reasons for these increases have been linked to a combination of climate change and past land use histories with general agreement that moist, high elevations forests are subject to climate-induced changes while drier, lower elevation forests are subject to combinations of climate and land use changes including grazing and fire exclusion policies (Agee, 1997; McKenzie *et al.*, 2004; Schoennagel *et al.*, 2004; Collins *et al.*, 2006; Westerling *et al.*, 2006; Littell *et al.*, 2009). Dillon *et al.,* (2011) used satellite-derived burn severity data to suggest that topography and climate were the principle influences on fire severity, and that only the southwestern U.S. experienced increases in area burned between 1984 and 2006. From 2000 to 2005, the average annual acreage burned was 70 percent greater than the acreage burned during the 1990s with annual fire management and suppression appropriations to federal land management agencies exceeding $3 billion (GAO, 2007). There was an average increase of over 283,000 hectares in annual total area burned on Forest Service administered lands during the period 1987 to 2002 compared to the period 1970 to 1986 (Calkin *et al.*, 2005). Westerling *et al*. (2006) associated the shift in annual area burned and large fire frequency during the mid-1980s with increased spring and summer temperatures and earlier spring snowmelt. Lightning-ignited fires have resulted in significant increases in hectares burned with no change in the number of lightning ignitions in the northern Rocky Mountains, southwest, and northeast (Stephens, 2005).

Although fire and bark beetles are both important drivers of vegetation dynamics in western North American forests, relatively few studies have addressed questions regarding their potential relationships. The earliest research used qualitative assessments of the potential effect

of mortality on subsequent fires using basic principles of fire ignition and spread (Brown, 1975; Knight, 1987). Later retrospective studies used combinations of fire history and remote sensing technologies to assess interactions in a more quantitative way (Bebi *et al.*, 2003; Kulakowski *et al.*, 2003; Bigler *et al.*, 2005; Kulakowski and Jarvis, 2011). Only recently have detailed stand level bark beetle effects and fire behavior potential been documented (Schulz, 2003; Romme *et al.*, 2006; Kulakowski and Veblen, 2007; Page and Jenkins, 2007a; Klutsch *et al.*, 2009; Jorgensen and Jenkins, 2011; Simard *et al.*, 2011).

In this paper, we critically review the influence of variably flammable, bark beetle-altered complexes of surface, ladder, and canopy fuels and the challenges of understanding the effects on extreme fire behavior, including the initiation and spread of crown fires in conifer forests in the Interior West. We will draw on examples from previously published work and combine and interpret results from various fire behavior modeling software systems and decision support tools to make general inferences about bark beetle-altered fuel complexes. The theoretical relationships presented are meant to draw the reader's attention to potential bark beetle, fuels and fire interactions, with an emphasis on the general nature of the ecological and physical processes at work. The inherent limitations, knowledge gaps and research needs in predicting and understanding these interactions will be discussed as well as potential future advances in overcoming these limitations. Our discussion ends with an emphasis on forest health as a measure of resistance and resilience to disturbance and the forest's ability to satisfy management objectives from both a planning and operational perspective. Specifically we will discuss:

- bark beetle effects on forest structure, composition and fuel bed characteristics,
- crown fire initiation and spread,
- challenges in modeling crown fire behavior in bark beetle-affected forests,

- short-and long-term implications for fire suppression considerations, and
- bark beetles, fires and forest health.

2. Bark Beetle Effects on Forest Structure, Composition and Fuel Bed Characteristics

The earliest and best known wildfire incident linked to bark beetle activity was the Sleeping Child Fire that occurred in western Montana in August 1961 (Fig. 1). At the time, it was the single largest wildfire in the US northern Rocky Mountain region in more than 20 years and "was treated as an event almost without precedent" (Lyon, 1984). The fuel accumulation resulting from a mountain pine beetle attack some 30 years earlier greatly increased the difficulty of controlling the fire (Roe *et al.*, 1971) as shown in the 1962 USDA Forest Service fire control training film *Fire Weather*.

It's only been relatively recently that research has systematically focused on an understanding of bark beetle, fuel and fire interactions (Jenkins *et al.*, 2008) in whitebark pine (Jenkins, 2011), Douglas-fir (<u>*Pseudotsuga menziesii*</u> (<u>Mirb.</u>) <u>Franco</u>) (Jenkins *et al.*, 2008), lodgepole pine (*Pinus contorta* Dougl) (Page and Jenkins, 2007a, b; Kaufmann *et al.*, 2008; Klutsch *et al.*, 2009; Klutsch *et al.*, 2011; Simard *et al.*, 2011; Schoennagel *et al.*, 2012), and Engelmann spruce (*Picea engelmannii* Parry ex Engelm) (Bebi *et al.*, 2003; Kulakowski *et al.*, 2003; Jorgensen and Jenkins, 2011). Much of this information is provided in an internet-based decision support tool created to assist fire managers and others interested in fuels and fire behavior in relation to bark beetle activity (Jenkins *et al.*, 2011).

The bark beetle rotation begins when a stand becomes susceptible to bark beetle infestation and is capable of supporting an outbreak or epidemic. Prior to the epidemic phase the bark beetle populations are considered endemic with only one to several trees attacked per

hectare (Bentz and Munson, 2000). The occurrence of epidemics often coincides with periods of short-term stress, such as drought (Berg *et al.*, 2006; Negrón *et al.*, 2009). Under stressful conditions, aggressive bark beetle species, like mountain pine beetle and spruce beetle, can overcome host tree resistance resulting in rapidly increasing population numbers. During the epidemic phase 80 percent or more of susceptible trees are killed. As time progresses, canopy openings result in significant increases in live shrub and herbaceous cover and loading and regeneration (Reid, 1989; Stone and Wolfe, 1996). In epidemic stands, the total amount of available canopy fuel is greatly reduced leading to decreases in canopy sheltering of wind and solar radiation (Brown, 1975; Knight, 1987). The length of the epidemic phase varies with conifer species, but generally lasts 5 to 10 years, and ends when most large diameter trees have been killed and the bark beetle population returns to endemic levels (Schmid and Amman, 1992). At this time, stands enter the post-epidemic phase which is characterized by the fall of dead trees. This phase lasts for decades to centuries until small surviving or newly regenerated host trees again reach susceptible age and size (Fig. 2).

The general changes in fuel bed characteristics over the course of a bark beetle rotation were described by Jenkins *et al.* (2008). In Figure 3 we illustrate the changes to forest structure, fine surface fuel and the abundance and condition of canopy fuels during the course of a bark beetle rotation in Engelmann spruce in central Utah. In general, there is a reduction in the number of live trees per hectare, average stand diameter, canopy base height, and the quantity, and quality of canopy foliage. During early stages of the outbreak there is an increase in the amount of dead canopy foliage and in turn a transfer of needles and some fine twig material from the conifer canopy to the forest floor such that surface litter amounts increase at the expense of canopy fuels (Page and Jenkins, 2007a; Jorgensen and Jenkins, 2011; Simard *et al.*, 2011). Total

surface fuel accumulation amounts to nearly a one-to-one transfer of aerial needles to surface litter, minus whatever decomposition occurs over the period of needle shed (Bigler and Veblen, 2011). Overstory tree mortality not only results in a pulse of needle litter and small diameter woody fuels to the forest floor during the epidemic phase, but also a release of shrubs and forbs in the early part of the post-epidemic (Reid, 1989; Stone and Wolfe, 1996; Jorgensen and Jenkins, 2011). The accumulation of coarse woody fuels is dependent upon the rate at which dead overstory snags fall to the surface. In Figure 4 the accumulation of coarse woody surface fuels is shown for three different rates of tree fall over the 100 years following spruce beetle outbreak. There is considerable variation in coarse woody fuel accumulation depending on stand structure, species composition, site, soils, physiography, and incidence of root disease and when compared to other bark beetle/host ecosystems (Mielke, 1950; Hinds *et al.*, 1965; Schmid and Hinds, 1974).

Jenkins (2011) described the characteristic changes of the crowns of individual attacked whitebark pine trees during the period of *Dendroctonus* bark beetle colonization, brood development and dispersal which is generally similar for other conifer species. In summary, an otherwise healthy, susceptible host tree retains a typical green (G) crown. During the first season of bark beetle infestation adults and early instar larvae are present within the inner bark and blue stain development has likely begun. Blue stain is caused by a complex of fungi that are carried in bark beetle mycangia (specialized mouthpart structures) and inoculated into the sapwood. The fungi spread in the sapwood through living parenchyma cells and the bordered pit pairs of water conducting tracheid elements. The degree of blue stain development is dependent upon degree of host colonization, fungal pathogenicity, host resistance and the ability of the tree to compartmentalize the fungi (Raffa and Berryman, 1983). The amount of sapwood affected thus

varies considerably, but in any case, infection reduces water flow to the crown resulting in a net reduction in the moisture content of the needle foliage (Parmeter *et al.*, 1989) and a drop in sapwood moisture (Reid, 1961). At this time, the crown is considered to be green-infested (Gi) (Wulder *et al.*, 2006). Larvae overwinter within the inner bark and at the beginning of the following season resume development. Rates of larval development vary depending on temperature, with pupation occurring within one-to-three years. Following pupation new adults emerge from the brood tree to colonize another susceptible host. In about twelve months after initial attack the tree crown will begin to fade to yellow (Y) (Amman, 1982), a transition largely regulated by blue stain development. In 16 months after attack, Y will have turned red (R). The R crown class can last up to 48 months after attack, but the time period varies with conifer host. Needle-fall accelerates during the R stage until all needles have fallen from the crown to the forest floor and the tree appears largely gray (Gr) (Wulder *et al.*, 2006). Many factors may influence crown change including the variability in bark beetle life histories and attack severity (number of successful beetle attacks per unit of bark surface area), the physical characteristics of host trees, and foliar moisture content and chemistry (Safranyik, 2003). Stand composition and structure, site and patterns of tree mortality influence the distribution of these classes and across the landscape and consequently the spatial distribution and arrangement of surface and canopy fuels.

3. Crown Fire Initiation and Spread

High intensity, stand-replacing crown fires are a common feature in certain conifer forest types in western North America (Arno, 2000; Baker, 2009), with or without disturbance-altered

surface and canopy fuels. In discussing the subject of tree mortality-caused fuel buildup in lodgepole pine forests, Brown (1975: 446) said:

> *The large increase in ground fuel and associated increase in the probability of large,*
> *high-intensity fires due to beetle epidemics suggests that the relationship among beetles,*
> *fire, and lodgepole pine tends to perpetuate lodgepole pine. The mountain pine beetles'*
> *strong preference for large trees gears heavy fuel buildup to a time when stands are*
> *mature or overmature. In some areas, this is when climax species are developing*
> *prominence in the understory and together with the ground fuels present a high chance of*
> *crown fire.*

Crown fires can also readily occur during the R stage of bark-beetle affected forests (Fig. 5) as vividly demonstrated recently during a major run of the Salt Fire on the Salmon-Challis National Forest in central Idaho on August 29, 2011 (Church *et al.*, 2011)[1]. Figure 6 illustrates the important fuel complex characteristics affecting crown fire dynamics. The most important parameter to consider in assessing crown fire potential is a surface fire's energy release rate or intensity. (Byram, 1959) defined fireline intensity (I, kW m^{-1}) as the rate of heat released from a linear segment of the fire perimeter as calculated by the following equation:

$$I_B = \frac{H \cdot w_a \cdot R}{60}$$

where H is the low heat of combustion (kJ kg^{-1}), w_a is the 'available fuel' or fuel consumed in the active flame front (kg m^{-2}), and R is the rate of fire spread (m min^{-1}). Flame length is its main visual manifestation (Alexander and Cruz, 2012).

The pioneering research of Van Wagner (1977) provides the fundamental basis for most current operational systems for predicting crown fire behavior although other models and

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[1] To view video footage taken by the USDA Forest Service of the Salt Fire taken on August 29, 2011 go to: http://www.youtube.com/watch?v=KKpBqdf16rE

modeling systems are slowly being produced elsewhere (e.g., Alexander and Cruz, 2011). Van Wagner's (1977) models do not provide for the prediction I_B or R, which must be derived by other means, but rather they provide quantitative criteria for determining the onset of crowning and active crown fire propagation.

Van Wagner (1977) developed a simple model for determining crown fire initiation on the basis of two canopy fuel properties:

$$I_o = (C \cdot \text{CBH} \cdot h)^{1.5}$$

where I_o is the critical surface fire intensity needed for initial crown combustion (kW m^{-1}), C is the criterion for initial crown combustion (kW$^{2/3}$ kJ^{-1} kg m$^{-5/3}$) which he described as "an empirical constant of complex dimensions," CBH is the canopy base height (m), and h is the heat of ignition (kJ kg^{-1}). The heat energy required to raise the crown foliage to its ignition temperature is in turn calculated using the following function (from Van Wagner, 1993):

$$h = 460 + 25.9 \cdot \text{FMC}$$

where FMC is the foliar moisture content (% oven-dry weight basis). The onset of crowning is thus expected to occur when CBH and FMC are sufficiently low for a surface fire of given intensity to ignite the foliage (i.e., when $I_B \geq I_o$). Vertical spread of fire into the crowns is for practical purposes assumed to be independent of the canopy bulk density (CBD, kg m^{-3}) which in turn represents the available canopy fuel load (ACFL, kg m^{-2}) divided by the depth of the canopy fuel layer (i.e., the average stand height less the CBH). Van Wagner (1977) also proposed a simple model for determining the requirement for a fully developed crown fire to occur in relation to the forest stand structure:

$$R_o = \frac{S_O}{CBD}$$

where R_o is the critical minimum spread rate for active crown fire (m min^{-1}) and S_o is the critical mass flow rate for solid crown flame (kg m^{-2} min^{-1}). Active crowning thus occurs when $I_B \geq I_o$ and the R after the onset of crowning is $\geq R_o$. Passive crowning occurs in cases where $I_B \geq I_o$ but $R < R_o$ (i.e., crown fuel consumption takes place but crown-to-crown spread is limited).

It is worth noting that in Van Wagner's (1977) crown fire theories, both passive and active crown fires are dependent on surface fire for their continued existence. The extent to which the intercrown distance (ICD) directly affects active crown fire propagation is presently unknown as this fuel complex characteristic is embedded in the CBD. However, from observation evidence obtained through experimental burning, it appears that active crown fires are able breach gaps in the forest canopy of 10-20 m with ease (Alexander *et al.*, 1991).

The value of the Van Wagner's (1977) equations is their simplicity, but that is also a major limitation as discussed by Alexander and Cruz (2011). For example, while Van Wagner's (1977) two equations are theoretically based, empirically-derived values are needed for the C and S_o quantities (currently given as 0.010 and 3.0 respectively). These values are presently based upon experimental fires in a red pine (*Pinus resinosa* Aiton) plantation in eastern Canada involving live, green needles (Van Wagner, 1968) for a specific set of conditions (i.e., FMC = 100%, CBH = 6 m, and CBD = 0.23 kg m^{-3}) (Cruz and Alexander, 2010).

4. Challenges in Modeling Crown Fire Behavior in Bark Beetle-Affected Forests

An increasing body of research and field observation is emerging describing the influence of bark beetle mortality on fire behavior. Various fire behavior prediction modeling systems (Cruz and Alexander, 2010) including BehavePlus in lodgepole pine (Page and Jenkins, 2007b; Schoennagel *et al.*, 2012), NEXUS in lodgepole pine (Simard *et al.*, 2011), and the Fuels and

Fire Extension (FFE) to the Forest Vegetation Simulator (FVS) in Engelmann spruce (DeRose and Long, 2009) and lodgepole pine (Klutsch *et al.*, 2011) have been used in assessing potential crown fire behavior in bark beetle-affected fuels.   At the heart of all these modeling efforts is the semi-physical surface fire model developed by Rothermel (1972) which is linked directly, or through various reformulations, to Van Wagner's (1977, 1993) crown fire initiation and propagation equations and Rothermel's (1991) statistically-derived crown fire rate of spread model. Cruz and Alexander (2010) have described in detail the various limitations and biases inherent in these models both individually and in the techniques employed in linking them together. For example, it is uncertain whether the Rothermel (1972) model can predict the rate of spread, and in turn, the intensity of surface fires that would lead to the onset of crowning without fuel model calibration.

One the most fundamental assumptions in forecasting or predicting wildland fire behavior is that the fuels are continuous, uniform, and homogeneous (Albini, 1976; Rothermel, 1983).   The more the situation departs from this ideal, the more difficult it is to forecast or predict wildland fire behavior. The mixture of physically and chemically altered canopy fuel in bark beetle-affected conifer forests results in rapid and highly unstable spatial and temporal variation (Fig. 5) creating difficulties in using the common crown fire behavior models. Existing crown fire initiation, propagation and spread rate models were developed to predict fire behavior where the majority of available fuels are composed of green or otherwise live, healthy foliage. Canopy fuels of trees recently affected by bark beetles undergo very rapid changes in FMC and chemistry that in turn affect forest flammability (Jolly *et al.*, 2012). Significant changes in the moisture content of tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder) foliage affected by sudden oak death have been documented suggesting an increased likelihood of crown fire

ignition (Kuljian and Varner, 2010). It is presently unknown to what degree existing models are able to account for these factors when bark beetle-altered canopy fuels represent a significant proportion of the canopy.

4.1 Onset of Crowning

Subsequent research has shown that the empirical constant *C* contained within Van Wagner's (1977) crown fire initiation model to be a variable quantity dependent on certain factors such as ladder fuels and certain surface fire behavior characteristics (Cruz *et al.*, 2006b).

Thus, use of the model is undoubtedly inappropriate in assessing crown fire potential in tanoak forests impacted by sudden oak death due to the abundance of dead, elevated leaf material with very low FMC values (Kuljian and Varner, 2010). Application to mountain pine beetle attacked stands still requires some form of field verification, either through experimental burning and/or wildfire observations. The present indications are that extension of the model to very low FMC values to stands with a distinct gap between the surface fuels and CBH seem hardly realistic when examined in the light of flame length as opposed to I_o (Fig. 7).

To properly apply Van Wagner's (1977) crown fire initiation model to disturbance-altered canopy fuel one must assume that the burning characteristics (total heat content, rate of energy release, etc.) of live and variously altered (Gi, Y and R) foliage are equivalent. Current evidence suggests that live and dead foliar fuels of the same species have different flammability characteristics, in terms of heat of combustion, ignitability, and sustainability, due to differences in ether extractable content including fats, resins, oils, and terpenes (Hough, 1969; Philpot and

Mutch, 1971). However, it is yet to be determined whether the presence of flammable volatile foliage organic compounds (Ormeño *et al.*, 2009) directly influences crown fire initiation.[2]

Once a tree dies many of these organic compounds are likely to decrease and break down due to their inherent volatility (Tingey *et al.*, 1980). Hough (1969) found significant differences in heats of combustion for live versus dead fuels of various herbaceous species and wiregrass. Likewise, Philpot and Mutch (1971) and Susott (1980) emphasized the importance of volatile extractives from live conifer foliage on fuel flammability due to relatively high heats of combustion and low vaporization temperatures. Others have shown increases in heat of combustion of dead material as it decomposes (Golley, 1961; Daubenmire and Prusso, 1963). Edmonds (1980) suggested that the increase was due to higher lignin concentration in dead material. Lignin has higher heat content and decomposes more slowly than other cell wall materials. Thus, the assumption that dead and live crown fuels have equivalent burning characteristics is not currently supported, which suggests that Van Wagner's (1977) crown fire initiation model may be inappropriately applied to canopy fuel conditions common in bark beetle-affected stands in the Gi to advanced Gr stages as done, for example, by Simard *et al*. (2011).

In addition to the complications inherent in Van Wagner's (1977) empirical constant, the heat of ignition function included within his crown fire initiation equation may not adequately capture the effects of changes in volatile content in live versus dead foliage on ignition temperatures even though it was applied to FMC values as low as 56% (Van Wagner, 1993). His model assumes the ignition of all fuels occurs at 300°C, which has been shown by Owens *et.al.* (1998) and Ormeño *et al*. (2009) to vary due to the presence of volatiles such as monoterpenes.

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[2] This is the subject of a current research project supported by the Joint Fire Science Program (JFSP) entitled "The Influence of Fuel Moisture and Flammable Monoterpenes on the Combustibility of Conifer Fuels" (JFSP 11-1-4-16) under the leadership of the first author.

They demonstrated the potential contribution of terpene content on foliage and litter

flammability by showing decreased time to ignition and increased proportion of burned material

where terpene contents were high. Van Wagner (1977) did acknowledge the potential role of a

lower ignition temperature in his model and determined that even if the ignition temperature

dropped by 100°C it would only lower the calculated ignition energy of fuel at a FMC of 100%

by about 6%. However, in the formulation of his heat of ignition function he assumed that all of

the moisture in the fuel most be driven off before ignition can occur (Van Wagner, 1968), which

has subsequently been shown not to be the case (Pickett *et al.*, 2010).

4.2 Crown Fire Rate of Spread

The most commonly used crown fire spread models are empirical in nature and are thus

limited in their potential scope of inference to the conditions for which the underlying data were

derived. Van Wagner (1977) described the conditions for which active crown fire becomes

possible using the CBD based on the available canopy fuel to determine a minimum threshold

needed to sustain active crown fire spread. Later he advocated a theoretical foliar moisture effect

(FME) function for adjusting crown fire rate of spread (Van Wagner, 1993) that he had devised

earlier (Van Wagner, 1974).   It is unknown whether this FME term, which has yet to be

validated for magnitude of its relative effect (Van Wagner, 1998), can adequately capture the

effect of dead foliage with FMCs lower than 30% of oven-dry weight because it was initially

developed, although not exclusively, for use in canopy fuels with FMC levels in the range of

about 90 to 130% of oven-dry weight (Van Wagner, 1974).

Rothermel's (1991) crown fire rate of spread model is based on a statistical correlation

between a limited number of wildfire observations (*n* = 9) in the northern Rocky Mountains (that

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presumably did not involve significant amounts of beetle-affected crown fuels) with predictions

for a standard fuel model based on his surface fire model.   A more robust empirically-based

model developed by Cruz *et al*. (2005) has been extensively tested against experimental fires and

wildfires (Alexander and Cruz, 2006), although as Alexander *et al*. (2006) point out, it is not

appropriate for use in bark beetle-altered fuel complexes. Thus, it is unlikely that either the

Rothermel (1991) or Cruz *et al*. (2005) model will be able to sufficiently predict active crown

fire spread in forests recently affected by bark beetles (i.e., R stage) as a standalone method at

the present time, although no comparisons against wildfire observations in such fuel complexes

have been undertaken to date.

4.3 Looking Ahead

The development of empirically-based fire behavior prediction models derived from

conducting experimental fires (e.g., Stocks, 1987) such as undertaken for the Canadian Forest

Fire Behavior Prediction System (Wotton *et al.*, 2009) is not considered a realistic option at the

present time in the western United States. Increasingly, physically-based fire behavior models

are being formulated that have their basis in fundamental chemistry and physics of combustion

and heat transfer processes (Morvan, 2011). These models hold great promise in being able to

advance our theoretical understanding of wildland fire dynamics. The approach has the potential

to simulate fire behavior in three-dimensional form (Parsons *et al.*, 2011) based on the unique

physical and chemical properties associated with bark beetle altered fuel complexes (Hoffman,

2011). However, the capacity of these models to adequately describe crown fire behavior is still

open to question, given there has been very limited testing of model performance and more

importantly there has been no evaluation against any empirical crown fire dataset undertaken to

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date (Alexander and Cruz, 2011). On the other hand, a physical-based model for predicting crown fire initiation developed by Cruz *et al.*, (2006b) has in fact been evaluated for its performance, at least in healthy conifer forest stands (Cruz *et al.*, 2006a).

The greatest potential for assessing the immediate impact (i.e. first 5 years) of bark beetles on conifer forest fire behavior will likely involve a combination of simulation or numerical modeling, experienced judgment, and case study knowledge (Williams and Rothermel, 1992; Alexander, 2007) involving both wildfires (historic and present day), operational prescribed fires, and experimental fires (Alexander and Taylor, 2010).

5. Short- and Long-Term Implications for Fire Suppression

Concerns about the consequences of bark-beetle altered fuel complex amongst fire managers is not new (Morrison, 1964, 1968; Maupin, 1979). Recent observations by fire managers on fires in western conifer forests confirm the influence of bark beetle-altered surface and canopy fuels on fire behavior and fire suppression operations (Stiger and Infanger, 2011). Firefighters have reported experiencing extreme fire behavior in currently infested trees with prolific spotting occurring in areas with heavy surface fuel buildup (Church *et al.*, 2011). Additionally firefighters have noticed that mountain pine beetle-affected trees tend to break off at mid-tree and uproot more easily than other dead trees. Stiger and Infanger (2011) urge a higher level of vigilance for firefighters exposed to bark beetle-affected forests. The same can certainly be said for members of the general public as well (Alexander *et al.*, 2012).

Safe and effective control of wildfires involves a multitude of issues (Alexander, 2000). Alexander and Stam (2003) have discussed fire suppression considerations beyond crown fire behavior that may be affected in post bark beetle-altered ecosystems and that can still lead to

other aspects of extreme fire behavior. For example, trees in various stages of decay will exhibit sloughing bark, loose branch material of various sizes, and persistent cones in the canopy providing abundant material to be lofted into convection columns for short and long range spotting (Rothermel, 1994). Coupled with the readily available firebrand material in bark beetle-affected forests is the receptive fuelbeds. In this regard, the following appeared in the administrative fire analysis report on the 1961 Sleeping Child Fire:

> *About 75% of the original timber stand was on the ground. A young stand of lodgepole pine and alpine fir was growing as understory in the remaining live stand. The dead and down timber had decomposed to the point where it would ignite easily and burn intensely during dry weather. Fuels were highly vulnerable to flying sparks and radiant heat.*

Prolific spotting greatly reduces fire suppression effectiveness and puts constructed fuelbreaks at risk of being compromised (Agee *et al.*, 2000).

Increased resistance to control through increases in fireline construction time (Broyles, 2011), time for snag mitigation for crew safety, reduction in areas suitable for fire shelter deployment (Rothermel and Mutch, 1986) and the ability to secure fireline due to the large accumulations of down woody fuels associated with the older Gr stage are also important considerations. These large diameter fuels have significant burn out times which puts constructed firelines at risk of being compromised for long periods of time and heavy coverage levels if fire retardants are to be effective (George and Fuchs, 1991). Additionally, these large diameter fuels can cause significant delays in attaining full control due to the extensive mop-up work required. Large numbers of dead standing trees can cause significant firefighter safety concerns due to the dangers associated with the snags falling on firefighters, especially when weakened by fire, and from the dangers associated with snag mitigation (Leuschen and

Frederick, 1999). Heavy accumulations of woody fuels also increase total heat output requiring the construction of larger safety zones (Butler and Cohen, 1998) and creating escape routes (Beighley, 1995), further slowing fire suppression operations.

6. Bark Beetles, Fires, and Forest Health

A common approach to managing bark beetles in general forest areas involves sanitation and salvage in which susceptible, infested and dead trees are removed to reduce local population levels, decrease residual stand susceptibility, and derive some economic benefit (Jenkins *et al.*, 2008). A more proactive approach is thinning in advance of an outbreak to create stands of young, small diameter trees and microclimatic conditions which are less favorable for bark beetle infestation and population spread (Whitehead *et al.*, 2007). Limitations on the size of thinning treatments, however, often reduce their potential for success. Post-disturbance salvage logging may be socially, politically and economically justifiable when a harvesting infrastructure, including roads, mills and markets exists, and when the value of standing trees is sufficient to cover harvesting costs. Otherwise, bark beetle management activities may not be justified if they are not compatible with forest sustainability and inhibit the natural role of the insect in promoting ecosystem health and biodiversity (Lindenmayer and Noss, 2006; Schmiegelow *et al.*, 2006).

In 2003, the US Congress passed the Healthy Forest Restoration Act (HFRA) after an especially costly fire year in 2002. HFRA is designed to reduce fire risk while improving commercial value for forest biomass in ways that contribute to forest health by managing insect outbreaks and fire. The legislation supports ecosystem enhancement, sensitive species,

biodiversity and carbon sequestration. Management practices implemented under HFRA

generally have a hazardous fuels reduction objective.

Two more recent pieces of legislation have the potential to fundamentally alter the way in

which wildfires are managed on public lands. The Federal Land Assistance, Management and

Enhancement (FLAME) Act of 2009 established an account to pay for fighting large, complex

wildland fires. The legislation will provide a separate budget for fighting the largest fires, so that

adequate funding is available and agencies' land management functions are not shorted during

costly fire years by "borrowing" from other resource programs to cover fire suppression costs.

The FLAME Act required the Secretaries of Agriculture and Interior to develop a Cohesive

Wildland Fire Management Strategy (CWFMS) and initiate a collaborative process between

government and non-government agencies and devise solutions to wildland fire management

issues. CWFMS is being implemented in phases and addresses the associations between

wildfire, insects and climate change and recognizes that declining vegetative health across

landscapes is significantly responsible for increased costs and losses associated with catastrophic

wildfires (USDA and USDI, 2011).

In order to achieve the biodiversity and ecosystem resiliency goals outlined within the

CWFMS fire management activities will need to incorporate multiple objectives that preserve

important ecological processes and recognize the inherent variability in forested ecosystems. To

be effective, hazardous fuel treatments will need to incorporate fine scale ecological principles

implemented at the landscape scale (Dellasala *et al.*, 2004). In those forested ecosystems that are

affected by severe bark beetle activity, hazardous fuel treatments should place potential

treatment gains in context with the ecosystems in which they are planned. In ecosystems where

high intensity, stand-replacing crown fires are a part of the natural fire regime, such as lodgepole

pine forests, fuel treatments may have little impact or be ecologically undesirable (Turner *et al.*, 2003; Schoennagel *et al.*, 2004). In these forests the most successful treatment strategies will focus on removing those trees deemed most hazardous to the public or other values in the wildland-urban interface and to provide access to firefighting personnel. Additional treatments along potential future fire control points (e.g., major road networks and useful topographic features) will aid future fire suppression operations by increasing firefighter safety and decreasing resistance to control.

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FIGURE CAPTIONS

Fig. 1. View of the convection/smoke column associated with the 1961 Sleeping Child Fire on the Bitterroot National Forest in western Montana that spread through lodgepole pine stands that had previously sustained heavy mortality from a mountain pine beetle outbreak in 1928 to 1932 (Roe and Amman 1970). This lightning-ignited fire started on August 4 and in spite of rapid initial attack grew to nearly 60 ha in the first two hours following detection and then to 3640 ha within 24 hours (Morrison 1964). The fire continued to grow until August 13 when it was controlled after having covered more than 11,330 ha of upper montane and subalpine forest. Photo by Ernest Peterson, USDA Forest Service.

Fig. 2. An Engelmann spruce stand near Purple Lake on the Fishlake National Forest in central Utah taken in (a) 1902 and repeat photo from the same location taken in (b) 2002 showing the stand as it regenerated following an extensive spruce beetle outbreak in the 1920s.

Fig. 3. Engelmann spruce beetle condition classes shown during the course of the bark beetle rotation; a) endemic (EN), b) epidemic (EP) and c) post-epidemic (PE). Changes in I) stand structure, II) canopy fuel condition and III) surface fuel complex are shown (Jorgensen and Jenkins 2011).

Fig. 4. Fire and Fuels Extension to the Forest Vegetation Simulator (Reinhardt and Crookston 2003) was used to simulate changes in large woody fuels greater than 7.62 cm in diameter resulting from low (Case 1) moderate (Case 2) and high (Case 3) fall rates following the spruce beetle outbreak in the 1920s at Purple Lake, Utah.   Repeat photos (top to bottom) were taken in 1948, 1968, 1988 and 2010 are representative of Case 3 fall rate. Data are from Jorgensen and Jenkins (2011).

Fig. 5. Aerial view (top) of a surface fire slowly, but steadily spreading upslope in green white spruce (*Picea glauca* (Moench) Voss) forest and then (bottom) transitioning to active crowning upon reaching a patch of "red stage" mountain pine beetle-attacked lodgepole pine forest. This wildfire occurred in the central interior region of British Columbia, Canada, around midafternoon on August 18, 2003. Photos by M. Simpson, British Columbia Wildfire Management Branch.

Fig. 6. Graphic illustration of fuel properties affecting crown fire initiation and spread. CBH = crown base height, ICD = intercrown distance, CBDmax and CBDmin = maximum and minimum canopy bulk density, SFI = surface fire intensity and FL = flame length. In the illustration we represent a fire spreading from left to right and depict suggested fire behavior from low intensity surface fire (a), passive crown fire (b), active crown fire (c) and high intensity surface fire (d) for the different fuel properties under constant wind and slope. See text for a discussion of the influence of foliar moisture and chemical composition on crown fire initiation and spread.

Fig. 7. Critical (a) surface fire intensity (b) equivalent flame length (based on Byram (159) for crown combustion in a conifer forest stand as a function of canopy base height and foliar moisture content (FMC) according to Van Wagner's (1977) crown fire initiation model for the mean FMC values associated with tree crown condition classes of mountain pine beetle attack in lodgepole pine as determined by Jolly et al. (2012).
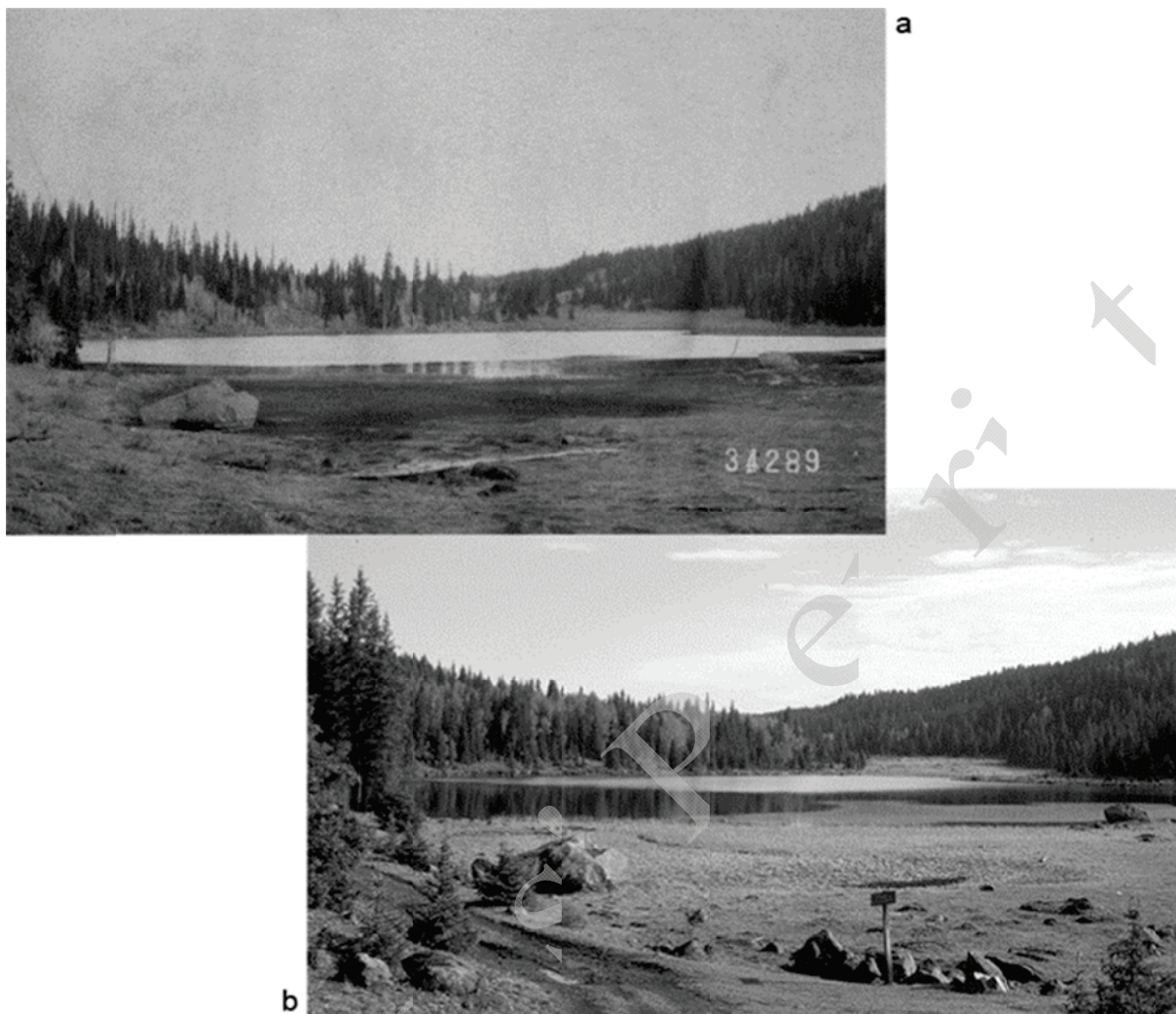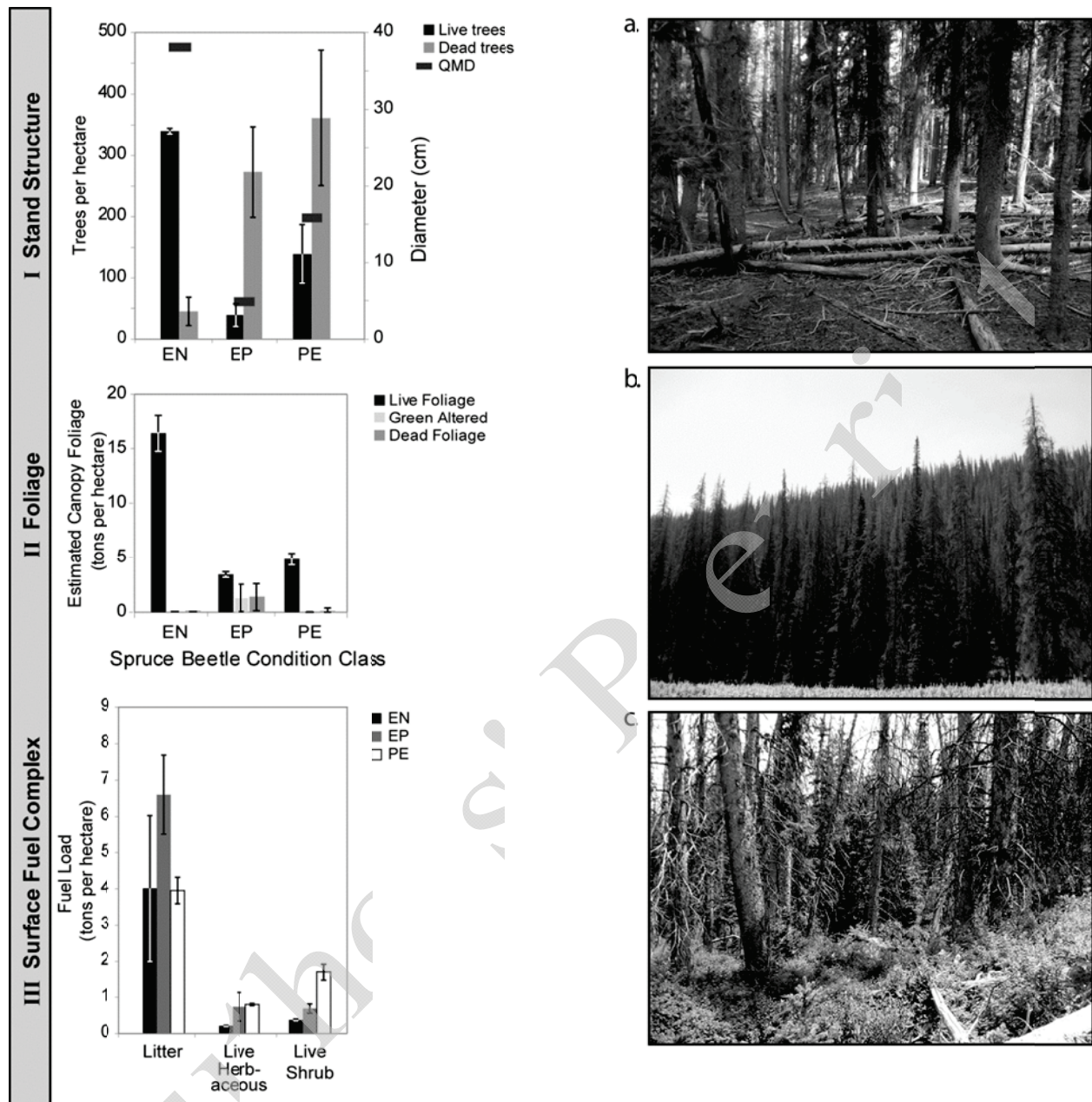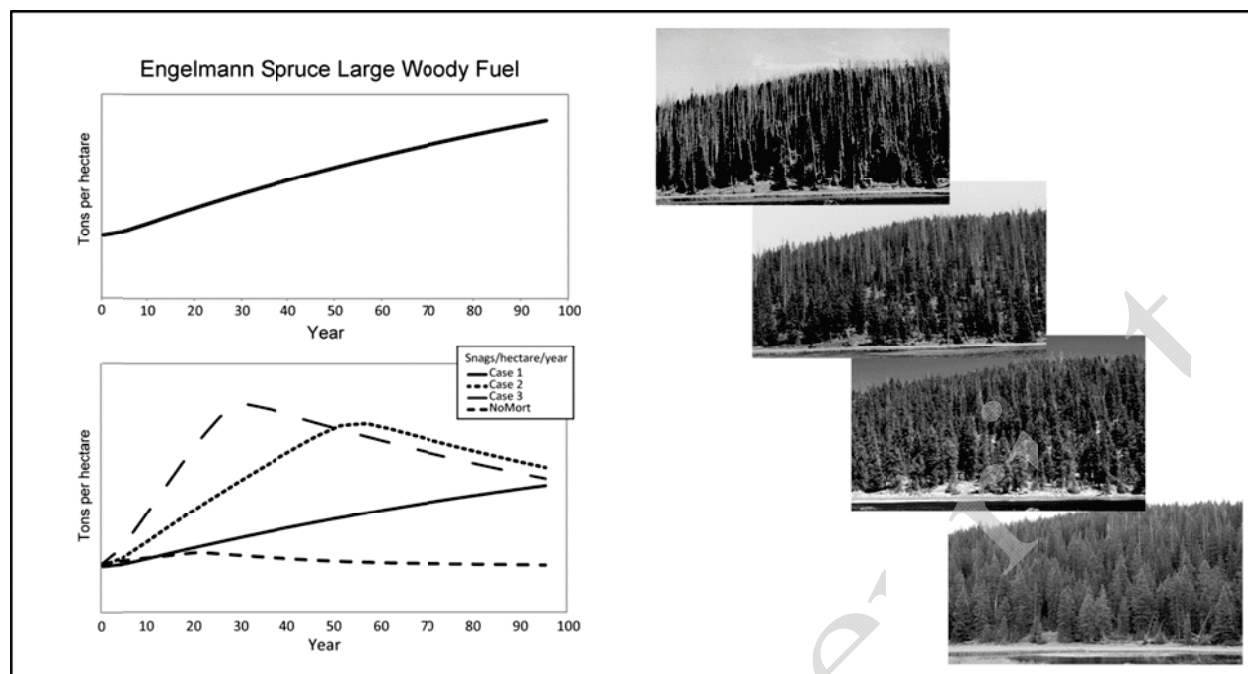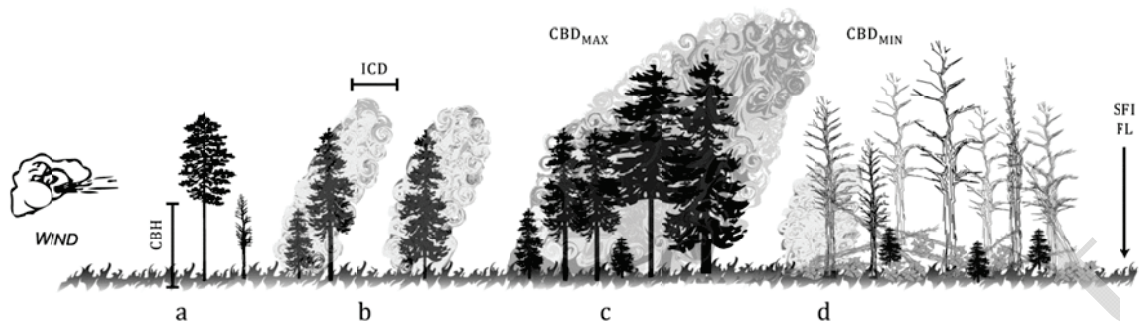
Figure 1

Figure 2

Figure 3

Engelmann Spruce Large Woody Fuel

Figure 4

Figure 5

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Figure 6

Figure 7